APPENDIX NOT INCLUDED

UPDATED SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT (SLERA) FOR THE GULFCO MARINE MAINTENANCE SUPERFUND SITE FREEPORT, TEXAS

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AET – apparent effects threshold

$AF_{soil/sediment}$ – chemical bioavailability factor from soil/sediment (unitless)

AST – aboveground storage tank

AUF – area-use factor (unitless)

BAF – bioaccumulation factor

BERA – Baseline Ecological Risk Assessment

BSAF – biota-sediment accumulation factor

BW – wildlife receptor body weight (kg)

C_{food} – chemical concentration in food (mg/kg)

$C_{soil/sediment}$ – chemical concentration in soil/sediment (mg/kg)

COI – chemicals of interest

COPEC – chemicals of potential ecological concern

CSM – conceptual site model

DDD – dichlorodiphenyldichloroethylene

DDE – dichlorodiphenyldichloroethane

DDT – dichlorodiphenyltrichloroethane

EPA – United States Environmental Protection Agency

EPC – exposure point concentration

ERA – Ecological Risk Assessment

ERL – effects range low

ERM – effects range medium

HPAH – high-molecular weight polynuclear aromatic hydrocarbon

HQ – hazard quotient

IR_{food} – food ingestion rate (kg/day)

$IR_{soil/sediment}$ – soil/sediment ingestion rate (kg/day)

LOAEL – lowest observable effects level

LPAH – low-molecular weight polynuclear aromatic hydrocarbon

NEDR – Nature and Extent Data Report

NOAEL – no observable adverse effects level

NPL – National Priorities List

PAH – polynuclear aromatic hydrocarbon

PCB – polychlorinated biphenyl

PCL – Protective Concentration Limit

QAPP – Quality Assurance Project Plan

RI/FS – Remedial investigation/Feasibility Study

RME – reasonable maximum exposure

ROPC – receptors of potential concern

SEL – Second Effects Level

SLERA – Screening-Level Ecological Risk Assessment

SMDP – Scientific Management Decision Point

SOW – Statement of Work

TCEQ – Texas Commission on Environmental Quality

TDSHS – Texas Department of State Health Services

TPWD – Texas Parks and Wildlife Department

TRV – species-specific toxicity reference value

TSWQS – Texas Surface Water Quality Standard

UAO – Unilateral Administrative Order

UCL – upper confidence limit

USDA – United States Department of Agriculture

USFWS – United States Fish and Wildlife Service

EXECUTIVE SUMMARY

The purpose and scope of this document is to summarize the analytical data for environmental media sampled during the RI and to conduct an updated Screening-Level Ecological Risk Assessment (SLERA) based on those data for the Gulfco Marine Maintenance Superfund Site located in Freeport, Texas in Brazoria County at 906 Marlin Avenue. The SLERA is a conservative assessment and serves to evaluate the need and, if required, the level of effort necessary to conduct a baseline ecological risk assessment. Per EPA guidance, the SLERA provides a general indication of the potential for ecological risk (or lack thereof) and may be conducted for several purposes including: 1) to estimate the likelihood that a particular ecological risk exists; 2) to identify the need for site-specific data collection efforts; or 3) to focus site-specific ecological risk assessments where warranted.

The Site consists of approximately 40 acres within the 100-year coastal floodplain along the north bank of the Intracoastal Waterway between Oyster Creek to the east and the Old Brazos River Channel to the west. Beginning in approximately 1971, barges were brought to the facility and cleaned of waste oils, caustics and organic chemicals, with these products reportedly stored in on-site tanks and later sold. Sandblasting and other barge repair/refurbishing activities also occurred on the Site. During the operation, wash waters were reportedly stored either on a floating barge, in on-site storage tanks, and/or in surface impoundments present on Lot 56 of the Site. The surface impoundments were closed under the Texas Water Commission's direction in 1982.

The South Area includes approximately 20 acres of upland that were created from dredged material from the Intracoastal Waterway. Prior to construction of the Intracoastal Waterway, this area was most likely coastal wetlands. The North Area, excluding the capped surface impoundments and access roads, is considered estuarine wetland. The North Area consists of approximately five acres of upland, which supports a variety of herbaceous vegetation that is tolerant of drier soil conditions, while the North wetlands is approximately 15 acres in size.

Data related to the nature and extent of potential contamination in ecologically-relevant media (e.g., soil, sediment, and surface water) at the Site were obtained as part of the Remedial Investigation. Unless otherwise noted, the samples were analyzed for the full suite of analytes as specified in the approved Remedial Investigation/Feasibility Study Work Plan for the Site. Samples included:

- Eighty-three surface soil samples (0 to 0.5 ft below ground surface) and 83 subsurface soil samples (0.5 ft to 4 ft below ground surface) were collected in the South Area.
- Eighteen surface soil and subsurface soil samples were collected in the North Area.
- Two additional surface soil samples were collected near the former transformer shed at the South Area for polychlorinated biphenyls analyses only.
- Ten background soil samples were collected within the approved background area approximately 2,000 feet east of the Site near the east end of Marlin Avenue.
- Sixteen sediment samples were collected from the Intracoastal Waterway in front of the Site. One additional sediment sample was collected near the Site and analyzed for DDT.
- Nine background sediment samples were collected from the Intracoastal Waterway east of the Site and across the main waterway canal.
- Forty-eight sediment samples were collected in the North Area wetlands. Additional sediment samples were collected from the North Area wetlands and analyzed for DDT; five of these samples were also analyzed for zinc.
- Eight sediment samples were collected from the two ponds located in the North Area.
- Four surface water samples were collected in the Intracoastal Waterway adjacent to the Site.
- Four surface water samples were collected from the background surface water area.
- Four surface water samples were collected in the North Area wetlands.
- Six surface water samples were collected from the two ponds located in the North Area.

All data were compared to appropriate ecological screening levels to identify the chemicals of potential ecological concern that were quantitatively evaluated further in the SLERA. Several representative groups of wildlife were identified as receptors of potential concern for use in the SLERA. Each group of receptors represents a group of species (ie., feeding guild) with similar habitat use and feeding habits that could potentially inhabit either the terrestrial, estuarine wetland, or aquatic habitats at the Site.

Potential ecological risks were calculated for the various representative species using a standard hazard quotient (HQ) approach for the various media using no observable adverse effect level-based toxicity reference values, high end exposure assumptions, and average and 95 percent upper confidence limits on the mean exposure point concentrations. In addition, a sample-by-sample analysis was also performed per EPA request (EPA, 2009a) to ensure that the sedentary benthic organisms were adequately protected.

Several of the risk calculations result in a reasonable maximum exposure (RME) HQ greater than one in soil from the South Area, North Area and background area. The RME HQs for antimony and zinc for the earthworm, Least shrew, and American robin in all three areas (i.e., South Area, North Area, and background area) were similar and ranged from 1.4 to 14.9. The RME HQ for 4,4'-DDD in the South Area was 1.2 for the earthworm receptor while the RME HQ for Aroclor-1254 in the South Area was 1.8 for the Least shrew. The RME HQs for the earthworm and American robin receptors in the background area were greater than 1 for barium as well.

The ERL-based HQs exceed one for 4,4'-DDT, several individual PAHs, total HPAHs, total PAHs, hexachlorobenzene, and gamma-chlordane for the benthic receptor in Intracoastal Waterway sediment. None of the HQs for the sandpiper or green heron exceed one. A sample-by-sample comparison shows that no compounds were measured in excess of their ERM in the Intracoastal Waterway; dibenz(a,h)anthracene was measured at a concentration greater than the midpoint of the ERL/ERM in one of sixteen samples; and hexachlorobenzene was measured in the same sample at a concentration greater than the AET, which was the only available benchmark for that compound.

The only compounds that exceeded their screening level in sediment from the background Intracoastal Waterway area were arsenic and nickel. None of the RME HQs for these two compounds or the other compounds exceeded one.

For the North Area wetlands sediment, the ERL-based HQs exceed one for 4,4'-DDT, a number of individual PAHs, total LPAHs, total HPAHs, total PAHs, endrin aldehyde, gamma-chlordane, and zinc for the benthic receptor. None of the HQs for the sandpiper or green heron exceed one. Measured concentrations of 2-methylnaphthalene (in 1 of 48 samples), acenaphthylene (in 2 of 48 samples), benzo(a)anthracene and benzo(a)pyrene (in 1 of 48 samples), chrysene (in 1 of 48 samples), gamma-chlordane (in 1 of 48 samples), phenanthrene (in 2 of 48 samples), pyrene (in 1 of 48 samples), total HPAHs (in 1 of 48 samples), and zinc (in 3 of 48 samples) exceed the midpoint of the ERL/ERM. Measured concentrations of dibenz(a,h)anthracene (in 5 of 48 samples), chrysene (in 1 of 48 samples), lead (in one of 48 samples), zinc (in 3 of 48 samples), and total HPAHs (in 3 of 48 samples) exceed their ERM.

The ERL-based HQs for dibenz(a,h)anthracene ranged from 3.2 to 17.4 for the average and RME benthic receptor scenarios, respectively, which suggest that adverse benthic risks from North Area wetlands sediment are possible. So, while localized adverse effects may be possible at the sampling locations that exceed the mid-point of the ERL/ERM and the ERM, it is difficult to estimate the potential significance of the impacts for the benthic community of the North Area wetlands, which is roughly 15 acres in size and is part of a wetlands system that covers hundreds of acres. Dibenz(a,h)anthracene is not considered bioaccumulative (TCEQ, 2001) and none of the risk estimates for the higher trophic-level receptors have HQs greater than one for this compound.

There are no indications that the benthic community in these six locations is stressed or has been impacted by the dibenz(a,h)anthracene or other compounds present in the sediment. It is unclear why the toxicity value for this compound is significantly lower than the benchmarks derived for the structurally similar PAHs but it is clear that this low value significantly impacts risk perception and should be taken in context with other benchmarks.

Lead and zinc were measured in at least one North Area wetlands sample at a concentration greater than the ERM, although their midpoint of the ERL/ERM HQs were less than one, and none of the NOAEL-based HQs for either compound were above one for the sandpiper or green heron.

The RME ERL-based HQs for 4,4'-DDT and zinc in pond sediment were greater than one, specifically 1.6 for 4,4'-DDT and 6.7 for zinc. The RME ERM-based HQ for zinc is 2.4 while the RME NOAEL-based HQs for both the sandpiper and green heron receptors exposed to zinc in pond sediments are 1.3 and the average HQs are 0.4. At all three sampling locations in the Small Pond, zinc was measured at a concentration greater than the ERM. Zinc concentrations measured in pond sediments at the Small Pond are similar to zinc measured in soil at the background area.

No compounds were measured in excess of their screening criteria in the Intracoastal Waterway surface water. Selenium (dissolved), which is considered bioaccumulative, was measured at a maximum and mean concentration roughly two and three times less, respectively, than the surface water quality standard. Even though selenium at the measured concentration range may not cause potential adverse effects to aquatic life, it is difficult to assess the likelihood of adverse risk to higher trophic-level receptors that prey on water-borne organisms.

Dissolved silver and 4,4'-DDT were measured in surface water from the background area of the Intracoastal Waterway above their ecological benchmark value or surface water quality standard in at least one sample. In addition, 4,4'-DDD, which is considered bioaccumulative, was detected in surface water from the background area of the Intracoastal Waterway but at a concentration below the benchmark. Both 4,4'-DDT and 4,4'-DDD are considered bioaccumulative. It is difficult to assess, however, the likelihood of adverse risk to higher trophic-level receptors that prey on water-borne organisms that may be exposed to these compounds.

Maximum concentrations of acrolein and dissolved copper were detected in at least one surface water sample from the wetlands area at concentrations that exceeded their respective ecological benchmark value or surface water quality standard. It is believed that there is insignificant risk from the presence of acrolein because of its infrequent detection (i.e., one of four samples) and the fact that its concentration is less than twice the ecological benchmark value. Although the maximum detected concentration of dissolved copper in a wetlands area surface water sample is greater than the Texas Surface Water Quality Standard (TSWQS) by about three-fold, the mean concentration of all four samples is less than the standard. Therefore, it was assumed that there is insignificant risk from the presence of copper in the wetlands area surface water. The maximum measured concentration of total mercury in the wetlands area is about 16 times less than the TSWQS. Even though mercury has bioaccumulative properties in aquatic and terrestrial food chains, it is believed that there is insignificant risk in the wetlands area because the maximum detected concentration is so much lower than the TSWQS.

The maximum concentration of dissolved silver detected in a surface water sample from the ponds exceeded its ecological benchmark value. The maximum and mean concentrations of dissolved silver in the ponds are about 15 times greater and 10 times greater, respectively, than the ecological benchmark value. However, the maximum and mean concentrations of dissolved silver in the surface water from the background area of the Intracoastal Waterway are more than twice the maximum and mean concentrations in the ponds. Additionally, the maximum and mean concentrations of dissolved silver in the surface water from the background area of the Intracoastal Waterway are about 31 times greater and 28 times greater, respectively, than the ecological benchmark value. Therefore, it is assumed that there is an insignificant risk from the presence of dissolved silver in the ponds.

Both selenium (total) and thallium (dissolved and total) were detected in surface water of the ponds at a concentration below the ecological benchmark. Both are considered bioaccumulative. It is believed that there is an insignificant risk in the ponds, however, because both were measured at concentrations so much lower than their ecological benchmark values.

1.0 INTRODUCTION

The United States Environmental Protection Agency (EPA) named the former site of Gulfco Marine Maintenance, Inc. (the Site) in Freeport, Brazoria County, Texas to the National Priorities List (NPL) in May 2003. The EPA issued a modified Unilateral Administrative Order (UAO), effective July 29, 2005, which was subsequently amended effective January 31, 2008. The UAO required the Respondents to conduct a Remedial Investigation and Feasibility Study (RI/FS) for the Site. The Statement of Work (SOW) for the RI/FS at the Site, provided as an Attachment to the UAO from the EPA, requires an Ecological Risk Assessment (ERA). The SOW specifies that the Respondents follow EPA's *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA, 1997). This guidance document proposes an eight-step approach for conducting a scientifically-defensible ERA:

1. Screening-Level Problem Formulation and Ecological Effects Evaluation;
2. Screening-Level Preliminary Exposure Estimate and Risk Calculation;
3. Baseline Risk Assessment Problem Formulation;
4. Study Design and Data Quality Objectives;
5. Field Verification of Sampling Design;
6. Site Investigation and Analysis of Exposure and Effects;
7. Risk Characterization; and
8. Risk Management.

Briefly, Steps 1 and 2 of the process are scoping phases of the ERA in which existing information is reviewed to preliminarily identify the ecological components that are potentially at risk, the chemicals of potential ecological concern (COPECs), and the transport and exposure pathways that are important to the ERA. This process is conducted using conservative assumptions to avoid underestimating risk or omitting receptors or COPECs, and constitutes the Screening-Level Ecological Risk Assessment (SLERA). Step 3 is the Baseline Problem Formulation that uses the results of the SLERA to identify methods for risk analysis and characterization, resulting in the identification of ERA data needs for the RI/FS. Steps 4 through 7 include formalization of the data needs, data collection, and data analysis for the risk characterization. Risk management activities are the eighth step in the process.

Steps 1 and 2 were performed through the submittal of an initial SLERA based on pre-RI data to EPA on November 17, 2005, as outlined in the SOW. The initial SLERA recommended collecting additional data to better characterize the nature and extent of contamination and potential risks associated with the Site. These data needs were identified in the RI/FS Work Plan (PBW, 2006a), which was approved with modifications by EPA on May 4, 2006 and finalized on May 16, 2006. Data needs were based on the preliminary conceptual site models (CSMs) provided in the Work Plan. Identification of COPECs for the baseline ecological risk assessment (BERA), which was one of the primary objectives of the initial SLERA, is based on maximum soil and sediment concentrations exceeding risk-based criteria. However, given the limited data available for the Site when the initial SLERA was conducted, eliminating COPECs from further evaluation or determining those that do required further evaluation could not be performed at that time.

As discussed at the August 4, 2005 Project Scoping Meeting and provided for in the RI/FS Work Plan, the SLERA and the resulting Scientific Management Decision Point (SMDP) were to be re-evaluated after the complete database of soil, sediment, and surface water samples collected during the RI was available. A Draft Nature and Extent Data Report (NEDR) providing these data was submitted to EPA on March 2, 2009 and was approved with modifications by EPA on April 29, 2009. The Final NEDR (PBW, 2009a), which incorporated the requested modifications, was submitted to EPA on May 20, 2009. This SLERA presents a re-evaluation of the November 16, 2005 SLERA (PBW, 2005), is based on the data presented in the NEDR (PBW, 2009a), and is responsive to EPA comments received on December 4, 2009 (EPA, 2009a) on the draft updated SLERA (PBW, 2009b).

1.1 PURPOSE AND SCOPE

The purpose and scope of this document is to summarize the analytical data for environmental media sampled during the RI and to conduct an updated SLERA based on those data. The SLERA is a conservative assessment and serves to evaluate the need and, if required, the level of effort necessary to conduct a baseline ecological risk assessment. Per EPA guidance (EPA, 2001), the SLERA provides a general indication of the potential for ecological risk (or lack thereof) and may be conducted for several purposes including: 1) to estimate the likelihood that a particular ecological risk exists; 2) to identify the need for site-specific data collection efforts; or 3) to focus site-specific ecological risk assessments where warranted.

This report provides documentation for whether further assessment (i.e., proceeding with the baseline ecological risk assessment) is necessary, and helps guide the next phases of evaluation, if necessary.

1.2 SITE SETTING AND HISTORY

The Site is located in Freeport, Texas in Brazoria County at 906 Marlin Avenue (also referred to as County Road 756). The Site consists of approximately 40 acres within the 100-year coastal floodplain along the north bank of the Intracoastal Waterway between Oyster Creek to the east and the Old Brazos River Channel to the west. Figure 1 provides a map of the site vicinity, while Plate 1 provides a detailed site map and shows site features and sampling locations.

During the 1960s, the Site was used for occasional welding but there were no on-site structures (Losack, 2005). According to the Hazard Ranking Score Documentation (TNRCC, 2002), from 1971 through 1999, at least three different owners used the Site as a barge cleaning facility. Beginning in approximately 1971, barges were brought to the facility and cleaned of waste oils, caustics and organic chemicals, with these products stored in on-site tanks and later sold (TNRCC, 2002). Sandblasting and other barge repair/refurbishing activities also occurred on the Site. At times during the operation, wash waters were stored either on a floating barge, in on-site storage tanks, and/or in surface impoundments on Lot 56 of the Site. The surface impoundments were closed under the Texas Water Commission's (TCEQ predecessor agency) direction in 1982 (Carden, 1982).

Marlin Avenue divides the Site into two areas. For the purposes of this report, it is assumed that Marlin Avenue runs due west to east. The property to the north of Marlin Avenue (the North Area) consists of undeveloped land and the closed surface impoundments, while the property south of Marlin Avenue (the South Area) was developed for industrial uses with multiple structures, a dry dock, sand blasting areas, an aboveground storage tank (AST) tank farm that is situated on a concrete pad with a berm, and two barge slips connected to the Intracoastal Waterway.

The South Area is zoned as "W-3, Waterfront Heavy" by the City of Freeport. This designation provides for commercial and industrial land use, primarily port, harbor, or marine-related activities. The North Area is zoned as "M-2, Heavy Manufacturing."

Adjacent property to the north, west and east of North Area is unused and undeveloped. Adjacent property to the east of the South Area is currently used for industrial purposes while the property directly to the west of the property is currently vacant and previously served as a commercial marina. The Intracoastal Waterway bounds the Site to the south. Residential areas are located south of Marlin Avenue, approximately 300 feet west of the Site, and 1,000 feet east of the Site.

2.0 SCREENING-LEVEL PROBLEM FORMULATION AND ECOLOGICAL EFFECTS EVALUATION (STEP 1)

Problem formulation establishes the goals, scope and focus of the SLERA by describing the physical features of the site, the communities of potential receptors present at the site, the selection of assessment and measurement endpoints, and potential exposure pathways. This information serves as the basis for the conceptual site model, which is used to focus the remaining steps of the SLERA.

2.1 ENVIRONMENTAL SETTING

The Site is located between Galveston and Matagorda Bays and is situated along approximately 1200 feet (ft.) of shoreline on the Intracoastal Waterway. The Intracoastal Waterway is a coastal shipping canal that extends from Port Isabel to West Orange on the Texas Gulf Coast and is a vital corridor for the shipment of bulk materials and chemicals. It is the third busiest shipping canal in the United States, and along the Texas coast carries an average of 60 to 90 million tons of cargo each year (TxDOT, 2001). Of the cargo carried between Galveston and Corpus Christi, TX, 49 percent is comprised of petroleum and petroleum products and 38 percent is comprised of chemicals and related products. Approximately 50,000 trips were made by vessels making the passage through the Intracoastal Waterway between Galveston and Corpus Christi, TX in 2006 (USACE, 2006).

The South Area includes approximately 20 acres of upland that were created from dredged material from the Intracoastal Waterway. Prior to construction of the Intracoastal Waterway, this area was most likely coastal wetlands. The North Area, excluding the capped impoundments and access roads, is considered estuarine wetland (USFWS, 2008). The North Area consists of approximately five acres of upland, which supports a variety of herbaceous vegetation that is tolerant of drier soil conditions, while the North wetlands is approximately 15 acres in size.

2.1.1 Terrestrial Areas

According to the United States Department of Agriculture (USDA) County Soils Maps (USDA, 1981), surface soils south of Marlin Avenue are classified as Surfside clays, and soils north of the road are classified as Velasco clays. Both soils are listed on the state and federal soils lists as hydric soils. The Velasco series consists of very deep, nearly level, very poorly drained saline soils. These soils formed in thick recent clayey sediments near the mouth of major rivers and streams draining into the Gulf of Mexico. They occur on level to slightly depressed areas near sea level and are saturated most of the year. Slope is less than one percent. The Surfside series consists of very deep, very poorly drained, saline soils that formed in recent clayey coastal sediments. They are saturated most of the year, and are on level to depressed areas near sea level with a slope less than one percent. It should be noted, however, that during drought periods, much of the wetlands area north of the Site is dry and desiccated, with standing water confined to very limited, localized areas.

Much of the South Area is covered with concrete slabs associated with former structures or Site operations. Because of the former industrial operations, the South Area contains very few areas of undisturbed terrestrial or upland habitat. Little resident wildlife has been observed at the South Area. During field work, nests were noted on some of the vertical structures at the Site.

The approximately five acres of terrestrial or upland habitat at the North area was created during previous operations at the Site. The five acres has developed some vegetation because plants have grown in some areas of the oyster-shell covered parking lot and former surface impoundments cap.

2.1.2 North Area Wetlands

There are two ponds on the North Area, located east of the former surface impoundments (Plate 1). The larger of the two ponds is called the Fresh Water Pond while the other pond is referred to as the Small Pond. It should be noted, however, that based on field measurements of specific conductance and salinity, the water in the Fresh Water Pond is brackish while water in the Small Pond is less brackish (but is not fresh water). The Fresh Water Pond water depth is generally 4 to 4.5 feet. The Small Pond is a shallow depression that tends to dry out during summer months and periods of drought; the water depth was approximately 0.2 feet when sampled in July 2006 and nearly dry when sampled in June 2008.

Based on field observations, the wetland in the North Area appears tidally influenced. Figure 2 depicts wetlands areas in the Site vicinity. Wetlands are the transitional zones between uplands and aquatic habitats and usually include elements of both. The wetlands at the Site are typical of irregularly flooded tidal marshes on the Texas Gulf Coast. The lower areas in the northern half of the property are dominated by obligate and facultative wetland vegetation such as saltwort (*Batis maritima*), sea-oxeye daisy (*Borrichia frutescens*), shoregrass (*Monanthocloe littoralis*), Carolina wolf berry (*Lycium caroliniaum*), spike sedge (*Eleocharis sp.*), and glasswort (*Salicornia bigelovii*). Higher ground near the road supports facultative wetland vegetation such as eastern bacchari (*Baccharis halimifolia*), sumpweed (*Iva frutescens*), and wiregrass (*Spartina patens*). Near Marlin Avenue, there are several shallow depressions that apparently collect and hold enough freshwater to allow homogenous stands of saltmarsh bulrush (*Schoenoplectus robustus*) to develop.

The high marsh, or supra-tidal zone, is the driest part of the coastal marsh habitat and supports far fewer invertebrate species. Due to the irregularity of flooding in the high marsh, there are no filter feeding bivalves or worms. Rather, the worms, amphipods, and isopods that live in the high marsh sediment are detritivores, direct deposit feeders, or predators. The crabs that live in the high marsh live in burrows that are excavated to groundwater, allowing them to keep their gills moist. Most crab species only return to the water to lay their eggs.

The North Area supports wildlife that would be common in a Texas coastal marsh. Fiddler crabs (*Uca rapax*) are likely the most abundant crustacean in the North Area. Other crustaceans found at the Site were fiddler crabs (*Uca panacea*), and hermit crabs (*Clibanarius vittatus*). The most common gastropod is the marsh periwinkle (*Littorina irrorata*). The Site is also used by a variety of shorebirds. Birds observed at the Site include the great blue heron (*Ardea herodias*), great egret (*Casmerodius albus*), snowy egret (*Egretta thula*), green heron (*Butorides striatus*), white ibis (*Eudocimus albus*), glossy ibis (*Plegadis falcinellus*), and willet (*Catoptrophorus semipalmatus*). The Site provides suitable habitat for rails, sora, and gallinules and moorhens, and may also be used by a variety of small mammals, rodents, and reptiles.

Other than gross disturbances in the wetlands area due to the former surface impoundment caps and other man-made upland terrain, the North Area wetlands is functionally and visually identical to the adjacent off-site wetlands area. Likewise, observations made during sediment sampling indicated consistent sediment characteristics for all North Area wetlands sampling locations.

2.1.3 Intracoastal Waterway

The Intracoastal Waterway supports barge traffic and other boating activities. The area near the Site is regularly dredged and, as noted by the United States Fish and Wildlife Service (USFWS), shoreline habitat is limited (USFWS, 2005a). Reduced light penetration, periodic dredging, wave action from barge traffic, and higher than normal tidal energy prevent submerged vegetation from growing in the Intracoastal Waterway near the Site. The absence of attached vegetation, which provides food and shelter, decreases the number of invertebrate species that can utilize the habitat in this sub-tidal zone and, therefore, most of the epibenthic invertebrates that utilize the sub-tidal zone in the Intracoastal Waterway near the Site are migrants.

Because of the reduced tidal energy at the upper end of each of the barge slips, there is a small amount of intertidal emergent marsh that has developed in these areas. Sand and silt has accumulated in the ends of the slips and is supporting small stands of gulf cordgrass (*Spartina alterniflora*). Sheetpile and concrete bulkheads protect the remainder of the shoreline. The bulkheads provide habitat for oysters (*Crassostrea virginica*), barnacles (*Balanus improvisus*), sea anemones (*Bunodosoma cavernata*), limpets and sponges.

Fishing has been known to occur on and near the Site. Red drum (*Sciaenops ocellatus*), black drum (*Pogonias cromis*), spotted seatrout (*Cynoscion nebulosus*), southern flounder (*Paralichthys lethostigma*) and other species are reportedly caught in the area (TPWD, 2009). It should be noted that, during the fish sampling conducted for the human health fish ingestion pathway risk assessment, red drum were not caught (using nets) as frequently as other species (see discussion in NEDR (PBW, 2009a)), presumably because of a lack of habitat and prey items to keep them near the Site. Recreational and commercial fishermen collect blue crabs (*Callinectes sapidus*) from waterways in the area. The Texas Department of State Health Services (TDSHS) has banned the collection of oysters from this area due to biological hazards and has issued a consumption advisory for king mackerel for the entire Gulf Coast due to mercury levels in the fish (TDSHS, 2005).

2.2 NATURE AND EXTENT OF POTENTIAL CONTAMINATION

Data related to the nature and extent of potential contamination in ecologically-relevant media (e.g., soil, sediment, and surface water) at the Site were obtained as part of the RI and, as noted previously, are discussed in the NEDR (PBW, 2009a). Unless otherwise noted, the samples were analyzed for the full suite of analytes as specified in the approved Work Plan (PBW, 2006a). Plate 1 provides sample locations for site-related samples, and Figure 3 provides sample locations for the background soil, surface water, and sediment samples. It should be noted on Plate 1, that different grid lines/areas and Zones 1 through 4 are identified. The grids were used to help locate samples based on EPA's preference to collect soil samples randomly over a grid while the zones represent the different areas where fish were sampled.

Tables 1 through 17 summarize the key parameters for the chemicals of interest (COIs) measured in these samples. A chemical of interest is defined in this report as any compound measured in at least one sample above the detection limit and at a detection frequency of greater than five percent. Tables 1 through 17 provide maximum and minimum measured concentrations, as well as summary statistics for each COI for each media. Average and 95% upper confidence limits (95% UCLs) on the mean were estimated using EPA guidance (EPA, 2002a) and are described in greater detail in the following section.

Eighty-three surface soil samples (0 to 0.5 ft below ground surface (bgs)) and 83 subsurface soil samples (0.5 ft to 4 ft bgs) were collected in the South Area. Eighteen surface soil samples and 18 subsurface soil samples were collected in the North Area. Two additional surface soil samples were collected near the former transformer shed at the South Area for polychlorinated biphenyls (PCBs) analyses only. Ten background soil samples were collected within the approved background area approximately 2,000 feet east of the Site near the east end of Marlin Avenue (Figure 3).

Sixteen sediment samples were collected from the Intracoastal Waterway in front of the Site. Nine background sediment samples were collected from the Intracoastal Waterway east of the Site and across the canal. One additional sediment sample was collected from the Intracoastal Waterway near the Site and analyzed for DDT to further characterize the extent of contamination as described in the NEDR (PBW, 2009a). Forty-eight sediment samples were collected in the North Area wetlands. Additional sediment samples were collected from the North Area wetlands and analyzed for DDT; five of these samples were also analyzed for zinc. A total of eight sediment samples were collected from the two ponds located in the North Area.

Four surface water samples were collected in the Intracoastal Waterway adjacent to the Site. Four surface water samples were collected from the background surface water area – the Intracoastal Waterway east of the Site, and across the canal (Figure 3). Four surface water samples were collected in the wetlands drainage areas north of Marlin Avenue and a total of six surface water samples were collected from the two ponds located in the North Area. Chemical analyses of these surface water samples included both total and dissolved concentrations of metals.

2.3 POTENTIALLY COMPLETE EXPOSURE PATHWAYS AND PRELIMINARY CONCEPTUAL SITE MODEL

The identification of potentially complete exposure pathways is performed to evaluate the exposure potential as well as the risk of effects on ecosystem components. In order for an exposure pathway to be considered complete, it must meet all of the following four criteria (EPA, 1997):

- A source of the contaminant must be present or must have been present in the past.
- A mechanism for transport of the contaminant from the source must be present.
- A potential point of contact between the receptor and the contaminant must be available.
- A route of exposure from the contact point to the receptor must be present.

Exposure pathways can only be considered complete if all of these criteria are met. If one or more of the criteria are not met, there is no mechanism for exposure of the receptor to the contaminant. Potentially complete pathways used in the SLERA are shown in the conceptual site models for the terrestrial and estuarine ecosystems (Figures 4 and 5, respectively).

In general, biota can be exposed to chemical stressors through direct exposure to abiotic media, or through ingestion of forage or prey that have accumulated contaminants. Exposure routes are the mechanisms by which a chemical may enter a receptor's body. Possible exposure routes include 1) absorption across external body surfaces such as cell membranes, skin, integument, or cuticle from the air, soil, water, or sediment; and 2) ingestion of food and incidental ingestion of soil, sediment, or water along with food. Absorption is especially important for plants and aquatic animals.

2.4 THREATENED AND ENDANGERED SPECIES

The USFWS was consulted (USFWS, 2005b) and information was obtained from the USFWS and Texas Parks and Wildlife Department (TPWD) regarding Threatened and Endangered Species. According to USFWS (USFWS, 2005c), Threatened and Endangered Species for Brazoria County include: bald eagle (*Haliaeetus leucocephalus*), brown pelican (*Pelecanus occidentalis*), green sea turtle (*Chelonia mydas*), hawksbill sea turtle (*Eretmochelys imbricate*), Kemp's ridley sea turtle (*Lepidochelys kempii*), leatherback sea turtle (*Dermochelys coriacea*), loggerhead sea turtle (*Caretta caretta*), piping plover (*Circus melodus*), and whooping crane (*Grus americana*). According to TPWD (TPWD, 2005), Threatened and Endangered Species for Brazoria County include: bald eagle (*Haliaeetus leucocephalus*), black rail (*Laterallus jamaicensis*), eastern brown pelican (*Pelecanus occidentalis*), interior least tern (*Sterna antillarum*), piping plover (*Circus melodus*), reddish egret (*Falco rufescens*), swallow-tailed kite (*Elanoides forficatus*), white-faced ibis (*Plegadis chihi*), wood stork (*Mycteria americana*), and corkwood (*Leitneria floridana*). None of these species have been observed at the Site but they are known to live in or on, feed in or on, or migrate through the Texas Gulf Coast and estuarine wetlands (TPWD, 2005).

2.5 ASSESSMENT AND MEASUREMENT ENDPOINTS

Assessment endpoints are explicit expressions of the ecological resource to be protected for a given receptor of potential concern (EPA, 1997). Identification of assessment endpoints is necessary to focus the SLERA on relevant receptors rather than attempting to evaluate risks to all potentially affected ecological receptors. Measurement endpoints comprise what are actually measured to protect the assessment endpoints. Assessment and measurement endpoints are discussed in relation to the risk question and testable hypotheses for each habitat and receptor group in Tables 18 and 19 (terrestrial and estuarine wetland/aquatic, respectively).

2.5.1 Terrestrial Assessment Endpoints

The terrestrial habitat associated with the Site includes the entire South Area and a small area of land adjacent to Marlin Avenue near the former surface impoundments in the North Area. The environmental value of this area is related to its ability to support plant communities, soil microbes/detritivores and wildlife. As indicated on Figure 4 and described in Table 18, the assessment endpoints for this area include:

- Vegetation survival, growth, and reproduction are values to be preserved in the terrestrial ecosystem. As food, plants provide an important pathway for energy and nutrient transfer from the soil to herbivores, omnivores, and invertebrates. Plants also provide critical habitat for terrestrial animals.
- Detritivore survival, growth, and reproduction and function (as a decomposer) are ecological values to be preserved in a terrestrial ecosystem because they provide a mechanism for the physical and chemical breakdown of detritus for microbial decomposition (remineralization), which is a vital function.
- Mammalian and avian herbivore and omnivore survival, growth, and reproduction are ecological values to be preserved in a terrestrial ecosystem because they are critical components of local food webs in most habitat types. In addition, small mammal and avian receptors can be important in the dispersal of seeds and the control of insect populations.
- Mammalian, reptilian, and avian carnivore survival, growth, and reproduction are values to be preserved in the terrestrial ecosystem because they provide food to other carnivores, omnivores, scavengers, and microbial decomposers. They also affect the abundance, reproduction, and recruitment of lower trophic levels, such as vertebrate herbivores and omnivores, through predation.

2.5.2 Estuarine Wetland and Aquatic Assessment Endpoints

The estuarine wetland habitat for the Site extends over the majority of the North Area while the Intracoastal Waterway (i.e., aquatic habitat) is south of the Site. Wetlands are particularly important habitat because they often serve as a filter for water prior to it going into another water body, they are important nurseries for fish, crab, and shrimp, and they act as natural detention areas to prevent flooding. The environmental value for these areas is related to their ability to support wetland plant communities, microbes/benthos/detritivores and wildlife. As indicated in Figure 5 and described in Table 19, the assessment endpoints for these areas include:

- Wetland vegetation survival, growth, and reproduction are values to be preserved in the estuarine wetland ecosystem. As food, plants provide an important pathway for energy and nutrient transfer from the soil to herbivores and omnivores as well as invertebrates. Plants also provide critical habitat for vertebrates and invertebrates.

- Benthos survival, growth, and reproduction are values to be preserved in estuarine ecosystems because these organisms provide a critical pathway for energy transfer from detritus and attached algae to other omnivorous organisms (e.g., polychaetes and crabs) and carnivorous organisms (e.g., black drum and sandpipers), as well as integrating and transferring the energy and nutrients from lower trophic levels to higher trophic levels. The most important service provided by benthic detritivores is the physical breakdown of organic detritus to facilitate microbial decomposition.
- Zooplankton survival, growth, and reproduction are values to be preserved in estuarine ecosystems. Zooplankton provide a food source for energy transfer through the water column-based pathway from phytoplankton to filter feeding and planktivorous organisms (e.g., finfish, shrimp, clams, worms, and oysters).
- Herbivorous and omnivorous fish and shellfish survival, growth, and reproduction are values to be preserved in estuarine ecosystems because they are critical components of the food web.
- Vertebrate carnivore (i.e., fish, fish-eating, and invertebrate-eating birds) survival, growth, and reproduction are values to be preserved in estuarine ecosystems. Vertebrates provide food for other carnivores and omnivores and affect species composition, recruitment, and abundance of lower trophic level organisms.

Because the Intracoastal Waterway is a deep, high-energy environment (i.e., dredged regularly) and light penetration is poor due to the high turbidity, submerged aquatic vegetation is not likely to thrive and, as such, is not an ecological resource to be protected as part of this assessment. Therefore, an assessment endpoint was not developed for submerged aquatic vegetation.

2.5.3 Measurement Endpoints

The measurement endpoints for the Site and the Intracoastal Waterway are the measurements of spatial distribution of chemical concentrations in soil, surface water and sediment to assess exposure concentrations for potentially exposed receptors. Maximum concentrations of chemicals measured in environmental media were compared to ecological benchmarks for the purposes of the screening-level problem formulation and ecological effects characterization (Step

1) of the SLERA. Food web dose calculations and comparisons with toxicity reference values as described in Section 3 provides a second measurement endpoint for higher trophic level receptors.

2.6 SELECTION OF AND COMPARISON TO ECOLOGICAL BENCHMARKS

This section describes the ecological benchmarks used to initially evaluate the data, and provides a summary of the comparison between Site data and the benchmarks. The benchmarks were chosen to conservatively represent the assessment endpoints since they are generally protective of the most relevant or sensitive endpoint for a variety of species. This was performed as an initial step in the SLERA process given the large number of analytes, media and receptors analyzed during the RI/FS and evaluated in the SLERA. It is believed that this is a reasonable step since the Site has been thoroughly characterized and the evaluation includes a robust data set. The COIs with no ecological benchmarks are discussed in the uncertainty section (Section 4.0).

It should be noted that any chemical considered to be bioaccumulative by the TCEQ (as defined in Table 3-1 of their ecological guidance document (TCEQ, 2006)) was retained for further evaluation if it was detected in at least one sample, even if it was reported below a screening criteria or if there was not a screening criteria. This approach was conservatively taken to ensure that food chain effects were considered for bioaccumulative compounds.

In addition, polynuclear aromatic hydrocarbons (PAHs) were evaluated as individual compounds, as a total concentration, and grouped as high-molecular weight (HPAH) or low-molecular weight (LPAH) as defined by TCEQ in Box 3-6 of the TNRCC (2001) ecological risk guidance. To quantitatively evaluate classes of PAHs in Step 2, individual PAHs were not eliminated from further assessment in Step 1 if it was detected in one sample of a given media, even if they were measured below their benchmark. It should be noted, however, if an individual PAH was not measured above the detection limit in any samples for that media, it was not included in the total PAH, HPAH, or LPAH estimate.

2.6.1 <u>Soil</u>

Soil sample data were compared with EPA and TCEQ ecological soil screening values contained in Tables 1 through 5. The EPA soil screening values were obtained from EPA's website at <u>www.epa.gov/ecotox/ecossl/</u> while the TCEQ values were obtained from Table 3-4 of TCEQ ecological guidance document (TCEQ, 2006). The screening value listed in Tables 1 through 5 is the lowest of the values provided by each Agency for plants, soil invertebrates, avians, and mammals (as indicated with the notation of "p", "i", "a", or "m", respectively).

South Area. Tables 1 and 2 provide a summary of the data for South Area soil samples. Only compounds with measured detections, including "J" flagged (or estimated) data, are listed in these tables. Table 1 contains only surface soil (0 to 0.5 ft bgs) data while Table 2 provides data for both surface and subsurface samples (0.5 ft to 4 ft bgs). This distinction was made to account for the different soil horizons that the different receptors may be exposed. For example, it was assumed that incidental ingestion of soil for the American robin would only occur within the 0 to 0.5 ft bgs soil whereas an earthworm may reasonably be exposed to the surface soil and the soil below 0.5 ft bgs as well. At least one South Area soil sample contained 4,4'-DDT, antimony, arsenic, barium, boron, cadmium, chromium, cobalt, copper, dieldrin, lead, lithium, manganese, mercury, molybdenum, nickel, vanadium, zinc, LPAHs or HPAHs at a concentration above an ecological benchmark. Figure 6 shows sample locations and associated concentrations of compounds measured above their screening value for select COPECs (because of the large number of exceedances, only those compounds that provides a hazard quotient greater than one in Section 3.0 were included in the figure). Although not reported in any South Area soil sample at a concentration above an ecological benchmark, 4,4'-DDD, 4,4'-DDE, Aroclor-1254, gamma-Chlordane, endrin aldehyde, and endrin ketone were detected in at least one South Area soil sample and are considered bioaccumulative in soil. These compounds, as well as those compounds with at least one sample concentration exceeding a benchmark, were evaluated further in the SLERA.

North Area. Tables 3 and 4 provide a summary of the data for North Area soil samples. Only compounds with measured detections, including "J" flagged (or estimated) data, are listed in these tables. Table 3 contains only surface soil data. Table 4 provides data for both surface (0 to 0.5 ft bgs) and subsurface samples (0.5 ft to 4 ft bgs). This distinction was made to account for the different soil horizons that the different receptors may be exposed. At least one sample contained antimony, barium, boron, cadmium, chromium, copper, dieldrin, lead, lithium, manganese, molybdenum, nickel, vanadium, zinc, or HPAHs at a concentration above its ecological benchmark. Figure 7 shows sample locations and associated concentrations of compounds measured above their screening value (because of the large number of exceedances, only those compounds that provides a hazard quotient greater than one in Section 3.0 were included in the figure). Although not reported in any North Area soil sample at a concentration above an ecological benchmark, endrin, endrin ketone, mercury, Aroclor-1254, 4,4'-DDE, and 4,4'-DDT were detected in at least one North Area soil sample and are considered bioaccumulative in soil. These compounds, as well as those compounds with measurements exceeding a benchmark, were evaluated further in the SLERA.

Background Soils. Table 5 provides a summary of the data for background soil samples (all surface samples). Only compounds with measured detections, including "J" flagged (or estimated) data, are listed in the table. At least one background sample contained antimony, barium, chromium, lead, lithium, manganese, zinc, or HPAHs at a concentration above its ecological benchmark. Figure 8 shows sample locations and associated concentrations of compounds measured above their screening value (because of the number of exceedances, only those compounds that provides a hazard quotient greater than one in Section 3.0 were included in the figure). Although not reported in any background soil sample at a concentration above the ecological benchmark, cadmium, copper, and mercury were detected in at least one background soil sample and are considered bioaccumulative in soil. These compounds, as well as those compounds with measurements exceeding a benchmark, were evaluated further in the SLERA. It should be noted that boron, strontium, titanium, and vanadium analyses were not performed on background soil samples.

2.6.2 Sediment

Sediment sample data were compared with EPA and TCEQ ecological screening values contained in Tables 6 through 9. The sediment screening values were the lower of the benchmark criterion obtained from EPA's ECO Update re: Ecotox Thresholds (EPA, 1996) and the TCEQ's ecological benchmarks listed in Table 3-3 of TCEQ (2006). The hierarchy for the benchmark values from the Ecotox Thresholds was marine sediment quality criteria, sediment quality benchmark, and Effects Range Low (ERL) value. It should be noted that the midpoint between TCEQ's Protective Concentration Limit (PCL) and Secondary Effects Level (SEL) is also presented in these tables for comparison purposes. In almost every case, if not for all cases, the PCL is the ERL and the SEL is the Effects Range Medium (ERM). As such, these terms were used interchangeably in the text, tables, and figures of the SLERA.

Intracoastal Waterway. Table 6 provides a summary of the data for sediment samples collected in the Intracoastal Waterway adjacent to the Site. Only compounds with measured detections, including "J" flagged (or estimated) data are listed in the table. At least one sample contained 4,4'-DDT, acenapthene, benzo(a)anthracene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, pyrene, fluoranthene, fluorene, phenanthrene, pyrene, LPAHs, HPAHS, or total PAHs at a concentration above an ecological benchmark. Figure 9 shows sample locations and associated concentrations of compounds measured above their screening value. Although not reported in any Intracoastal Waterway sediment sample at a concentration above an ecological benchmark, copper, gamma-Chlordane, hexachlorobenzene, mercury, nickel, and zinc were detected in at least one sediment sample and are considered bioaccumulative in sediment. These compounds, as well as those compounds with measurements exceeding a benchmark, were evaluated further in the SLERA.

Intracoastal Waterway Background. Table 7 provides a summary of the data for sediment samples collected in the Intracoastal Waterway background area. Only compounds with measured detections, including "J" flagged (or estimated) data, are listed in the table. At least one sample contained arsenic or nickel at a concentration above its ecological benchmark, as shown in Figure 10. Although not reported in any Intracoastal Waterway background sample at a concentration above an ecological benchmark, copper, 4,4'-DDT, mercury, and zinc were detected in at least one sediment sample and are considered bioaccumulative in sediment. These compounds, as well as those compounds with measurements exceeding a benchmark, were evaluated further in the SLERA.

Wetlands. Table 8 provides a summary of the data for sediment samples collected in the wetlands area north of Marlin Avenue. Only compounds with measured detections, including "J" flagged (or estimated) data, are listed in the table. At least one sample contained 2-methylnaphthalene, 4,4'-DDT, acenaphthene, acenaphthylene, anthracene, arsenic, benzo(a)anthracene, benzo(a)pyrene, chrysene, copper, dibenz(a,h)anthracene, endosulfan sulfate, fluoranthene, fluorene, gamma-chlordane, lead, nickel, phenanthrene, pyrene, zinc, LPAHs, HPAHs, or total PAHs at a concentration above its ecological benchmark. Figure 11 shows sample locations and associated concentrations of compounds measured above their screening value. Although not reported in any wetlands sediment sample at a concentration above an ecological benchmark, cadmium, endrin aldehyde, endrin ketone, and mercury were detected in at least one sediment sample and are considered bioaccumulative in sediment. These compounds, as well as those compounds with measurements exceeding a benchmark, were evaluated further in the SLERA.

Ponds. Table 9 provides a summary of the data for sediment samples collected in the ponds north of Marlin Avenue. Only compounds with measured detections, including "J" flagged (or estimated) data, are listed in the table. At least one sample contained 4,4'-DDT or zinc at a concentration above its ecological benchmark as shown in Figure 12. Although not reported in any pond sediment sample at a concentration above an ecological benchmark, cadmium, copper, 4,4'-DDD, and nickel were detected in at least one sediment sample and are considered bioaccumulative in sediment. These compounds, as well as those compounds with measurements exceeding a benchmark, were evaluated further in the SLERA.

2.6.3 Surface Water

Surface water samples were compared with national water quality criterion, Texas Surface Water Quality Standards (TSWQS), and TCEQ ecological screening criteria, which were obtained from TCEQ's ecological benchmarks listed in Table 3-2 of TCEQ (2006). If the benchmark was listed for dissolved concentrations (only applicable to metals), it was not compared to the total concentration data.

Intracoastal Waterway. Tables 10 and 14 summarize the analytical data for total and dissolved concentrations, respectively, for surface water samples collected from the Intracoastal Waterway adjacent to the Site. Since there were no compounds that were measured in excess of a screening level, there is not a figure to identify exceedances. Selenium, which is considered bioaccumulative in water, was measured in four of four surface water samples collected from the Intracoastal Waterway but at concentrations below the benchmark.

Intracoastal Waterway Background. Tables 11 and 15 summarize the analytical data for total and dissolved concentrations, respectively, for surface water samples collected in the Intracoastal Waterway background area, east of the Site and across the Intracoastal Waterway. Figure 13 shows sample locations and associated concentrations of compounds measured above their screening value. 4,4'-DDT and dissolved silver were detected in at least one sample in excess of their respective benchmark values. 4,4'-DDD and 4,4'-DDT were detected in two of four and one of four surface water samples collected at the background locations and are considered bioaccumulative although it should be noted that 4,4'-DDD was not measured at a concentration greater than the benchmark.

Wetlands. Tables 12 and 16 summarize the analytical data for total and dissolved concentrations, respectively, for surface water samples collected in the wetlands drainage areas north of Marlin Avenue. Acrolein and dissolved copper were detected in at least one sample in excess of their respective benchmark. Figure 14 shows sample locations and associated concentrations of compounds measured above their screening value. Mercury, which is considered bioaccumulative, was detected in two of four surface water samples (total concentrations only) but below a benchmark for a dissolved concentration.

Ponds. Tables 13 and 17 summarize the analytical data for total and dissolved concentrations, respectively, for surface water samples collected in the two ponds located in the North Area. Dissolved silver was detected in all six pond surface water samples in excess of its benchmark value. Figure 15 shows sample locations and associated concentrations of compounds measured above their screening value. Thallium, which is considered bioaccumulative by the TCEQ, was measured in all three dissolved surface water samples collected from the Small Pond. Selenium, which is also considered bioaccumulative in water, was measured in one total surface water sample collected from the Small Pond. No concentration of selenium or thallium was measured above their benchmarks.

2.7 COMPARISON TO THE BACKGROUND AREAS

Soil samples were collected at ten off-site locations; sediment samples were collected at nine off-site locations in the Intracoastal Waterway; and four surface water samples were collected at four off-site "zones" in the Intracoastal Waterway as described in the Work Plan (PBW, 2006a) to help provide an understanding of what COIs and concentrations may be considered site-related. This information was used to characterize Site conditions in the NEDR (PBW, 2009a).

EPA guidance for conducting SLERAs (EPA, 2001) recommends that comparison with background generally not be used to remove compounds from further evaluation in order to conservatively ensure that site risks are adequately characterized. This recommendation is based on the premise that the SLERA is often conducted on limited data set prior to a comprehensive site characterization. A background comparison, however, was conducted in this SLERA because: 1) a large Site data set was developed during the RI (including data for an approved and Site-specific background area); 2) the nature and extent of contamination at the Site has been thoroughly and completely characterized, and 3) the high quality of the Site and background data allows for a reliable comparison.

The soil background data were compared to soil from the South Area and North Areas of the Site, as well as sediments from the North wetland and the North Area ponds. As described in the NEDR (PBW, 2009a), based on similarities in composition and condition between background soil and sediments of the North wetlands area, this comparison was appropriate. Sediment and surface water data for the Intracoastal Waterway samples were compared to sediment and surface water data collected in the Intracoastal Waterway background location.

Comparisons between Site sampling data and Site-specific background data were conducted for all inorganic compounds measured in excess of their respective benchmark values. Background comparisons were also made for compounds considered bioaccumulative but measured at a concentration less than the benchmark. The background comparisons were performed in accordance with EPA's *Guidance for Comparing Background and Chemical Concentrations in Soil for CERCLA Sites* (EPA, 2002b). Distribution testing was conducted to estimate 95% UCLs and the summary statistics were used to perform comparison of the means analysis. The output of these background statistical comparison tests is provided in Appendix B. Table 20 summarizes the results of the testing and indicates whether the site data were found to be statistically different than the background data.

In several instances (e.g., lithium in South Area soil; barium in North Area wetlands sediment), statistical differences between the two data sets were due to higher concentrations in the background population, as noted in Table 20. It should be noted that no compounds were eliminated from further consideration in the SLERA based on the comparison to background concentrations. The list of COPECs carried through Step 2 of the SLERA is presented in Table 21 and includes any compound measured above its screening level in at least one sample, or any compound measured above its detection limit that is considered bioaccumulative per TCEQ guidance (TCEQ, 2006).

A statistical comparison between Site surface water and background surface water could not be conducted given the small size of both data sets. Visual inspection of the data indicates that there is no consistent observable difference between the data sets and COIs.

3.0 SCREENING-LEVEL PRELIMINARY EXPOSURE ESTIMATE AND HAZARD QUOTIENT CALCULATION (STEP 2)

The screening-level exposure and risk calculation description presented in this section of the SLERA corresponds to Step 2 of EPA guidance (EPA, 1997). Step 2 includes a quantitative assessment of potential ecotoxicity and the result of Step 2 is a decision on whether additional ecological risk evaluation is necessary.

3.1 RECEPTORS OF POTENTIAL CONCERN

Several representative groups of wildlife were identified as receptors of potential concern (ROPCs) for use in the SLERA. Each group of receptors represents a group of species (ie., feeding guild) with similar habitat use and feeding habits that could potentially inhabit either the terrestrial, estuarine wetland, or aquatic habitats at the Site. Representative species groups that may use the habitats at the Site are described briefly below. When several species may be present that could represent the feeding guild for a habitat, the species was chosen as the ROPC for that feeding guild based on its habitat affinity and potential for exposure. It should be noted, however, that each species chosen below as the representative receptor is symbolic of the entire guild so that all species within that guild are evaluated (and protected), not just the representative species/receptor.

3.1.1 Terrestrial Receptors

- Detritivores, Invertebrates and Terrestrial Plants. There are limited terrestrial areas at the Site. The earthworm was chosen to represent detritivores and invertebrates for the terrestrial ecosystem in this area because it is an important part of the food chain as prey for some first-order carnivores.

- Mammalian Herbivores and Omnivores. Habitat type plays a major role in the presence and abundance of the various species of mammals found at the Site. Of the three major groups of mammalian receptors (carnivores, ungulates, and rodents) potentially found at the Site, the small mammalian rodents are the most diverse and complex, and are most likely to have the highest area use factor. The habitat most likely does not support an ungulate population because it does not provide protective cover that they prefer although they may graze on some of the terrestrial plants on occasion. The omnivorous deer mouse (*Peromyscus maniculatus*) and Least shrew (*Cryptotis parva*) were selected as the ROPCs for the various feeding guilds of small mammals at the Site. Dietary composition for the deer mouse, with an assumed area use factor of 100 percent, was assumed to be 10% terrestrial invertebrates and 90% terrestrial plant tissue while the dietary composition for the Least shrew, with an assumed area use factor of 100 percent, was assumed to be 90% terrestrial invertebrates and 10% terrestrial plant tissue in order to assess the potential exposures to a receptor ingesting a general mix of prey types at the Site. The deer mouse was assumed to have a 2% incidental soil ingestion rate and the Least shrew was assumed to have an 8% incidental soil ingestion rate (EPA, 2009a).

- Mammalian Carnivores. Carnivores potentially present include omnivores such as the spotted and striped skunks, raccoon, and coyote (*Canis latrans*). A skunk was observed at the Site and fecal evidence of a carnivorous species was also observed at the Site. Since some of the COPECs are considered bioaccumulative compounds, assessing risks to an upper trophic level receptor is appropriate. Therefore, the coyote (*Canis latrans*) was selected as the ROPC for the mammalian carnivore feeding guild as it may feed at the Site on occasion as part of its larger home range. An area use factor of 100 percent was conservatively assumed per EPA (1997), and it was assumed that the coyote ingests 2% of its dietary intake via incidental soil ingestion (EPA, 2009a).

- Reptilian Carnivores. A representative reptilian predator for the Site is the rat snake (*Elaphe obsolete*), which has been observed at the Site. Rat snakes feed primarily on small mammals and eggs.

- Avian Herbivores and Omnivores. In general, avian species are influenced by the same types of landscape components as mammals, although vegetation is by far the more important factor. Birds are generally less important than mammals in terrestrial risk assessments because they live in less intimate contact with the soil, are highly mobile, and in many cases are present only seasonally. Most small birds have flexible diets that emphasize specific types of plant or animal material during certain seasons and most species are opportunistic, feeding on whatever food source is most abundant or particularly nutritious/palatable at a given time. A generalized avian receptor, represented by the American robin (*Turdus migratorius*), was selected to represent the omnivorous feeding guild. An area use factor of 100 percent per EPA (1997) and a 5.2% incidental soil ingestion rate per EPA (2009a) were conservatively assumed.

- Avian Carnivores. Representative avian predators (raptors) for the Site include the red-tailed hawk (*Buteo jamaicensis*) although it has not been observed at the Site. It, however, may use the Site for hunting prey occasionally. Red-tailed hawks feed primarily on small rodents, snakes, and lizards although they are opportunistic and will feed on other prey at times. An area use factor of 100 percent per EPA (1997) and a 2% incidental soil ingestion rate per EPA (2009a) were conservatively assumed.

3.1.2 Estuarine Wetland and Aquatic Receptors

- Benthos. Polychaetes burrow in and ingest sediment and have a greater exposure potential to sediment-bound chemicals than most epibenthos organisms such as shrimp and crab. Polychaetes are likely to be the most abundant class of benthic organisms found in the Intracoastal Waterway and, as such, *Capitella capitata* was chosen as the ROPC to represent this receptor class.

- Fish and Shellfish. Fiddler crabs (*Uca rapax*) and killifish (*Fundulus grandis*) were chosen as the ROPC to represent herbivorous or omnivorous species in the estuarine wetland and aquatic ecosystems, respectively. Fiddler crabs and their burrows are abundant at the Site. They eat detritus (dead or decomposing plant and animal matter) and serve as a food source for many wetland animals. It was assumed that their area use factor is 100 percent. The killifish was chosen to represent this feeding guild because it is likely to be present in the area of the Site and because it is an omnivorous fish that feeds primarily on organic detritus, small crustaceans, zooplankton, epiphytic algae, and polychaetes. Killifish may inhabit the Site for its entire life cycle; therefore, an area use factor of 100 percent was assumed.

- Carnivorous Fish. Black drum (*Pogonias cranius*) was selected as the first order carnivore ROPC because it is present in the Intracoastal Waterway and because it is an omnivorous carnivore that eats shrimp, crabs, small fish, benthic worms and algae. Per EPA (1997), an area use factor of 100 percent was conservatively assumed. The spotted seatrout (*Cynoscion nebulosus*) was chosen to represent a second order carnivorous fish species because it is present in the Intracoastal Waterway and because adult fish feed almost exclusively on other fish. It was conservatively assumed that the area use factor for the spotted seatrout is 100 percent per EPA (1997).

- Avian Carnivores. Sandpipers (*Calidris genus*) were chosen as first order avian carnivore ROPC because they have been observed at the Site. Although not observed at the Site, the green heron (*Butorides striatus*) was chosen as the second order avian predator ROPC to assess food chain impacts. Sandpipers are migratory birds that feed on aquatic insects and larva, marine worms, small crabs, small mollusks, and other invertebrate prey items. An area use factor of 100 percent was conservatively assumed per EPA (1997). Green herons are migratory birds that feed on small fish, invertebrates, insects, frogs, and other small animals. Per EPA (1997), an area use factor of 100 percent was conservatively assumed for green herons as well. Both were assumed to have an incidental sediment ingestion rate of 2% of dietary intake (EPA, 2009a).

3.2 SCREENING-LEVEL EXPOSURE ESTIMATES

In the exposure analysis, potential exposure of ecological receptors to COPECs was quantified. There are two basic routes of exposure for the COPECs and receptors at the Site: 1) ingestion from food and soil/sediment; and 2) direct contact. Quantification of exposure potential for both of these exposure routes requires data on chemical concentrations in environmental media (e.g., soil, sediment, prey items) and ingestion rates or contact information for each receptor and pathway. In addition, body weights, home range size, and other factors must be known for each of the receptors, as well as the chemical and physical properties of the COPECs.

Ecological receptors based on an ingestion pathway include birds, crustaceans, mammals, and fish. Receptors evaluated based on direct contact include earthworms in the terrestrial ecosystem and polychaetes and amphipods in the wetlands/aquatic ecosystem. Tables 22 and 23 provide exposure parameters for each receptor for terrestrial and estuarine wetland/aquatic receptors, respectively. In most instances, exposure parameters were chosen from regulatory or peer-reviewed literature and maximum ingestion rates and minimum body weights were preferentially used, when available. Best professional judgment was used when information for a ROPC was not available. References for the selected values are shown in the tables and the reference citations are included in Section 6.0.

Exposures via inhalation or dermal absorption were not evaluated for most receptors because of a lack of appropriate exposure and toxicity data and the uncertainty associated with these pathways (TNRCC, 2001). The exposure of animals to contaminants in soil by dermal contact is likely to be small due to barriers of fur, feathers, and epidermis. Therefore, the SLERA focused on the ingestion pathways as the primary exposure route for all vertebrates (unless direct contact was specifically noted and assessed).

For most receptors evaluated based on ingestion, exposure was quantified by estimating the daily dose (mg COPEC/kg body weight per day) that the receptor is expected to receive via both incidental soil/sediment ingestion and through dietary intake from food items and prey. For the direct contact pathway (i.e., earthworm and polychaetes), the COPEC concentration in soil or sediment was used directly to estimate exposure.

EPA guidance (EPA, 1997) suggests conservatively using maximum concentrations in the SLERA, which is often performed when only limited data sets are available. During the scoping meeting with EPA, it was discussed that a 95% upper confidence limit (UCL) on the average concentration would more appropriately represent the exposure point concentration (EPC) given the extensive characterization and sampling that has been conducted at the Site during the RI. The general procedure that is recommended by EPA to estimate a 95% UCL (EPA, 2002a) was used as the EPC to represent the upper end of exposure. EPA's ProUCL Version 4 program (EPA, 2007a) was used to analyze dataset distribution and calculate average and 95% UCL concentrations. ProUCL calculates various estimates of the 95% UCL of the mean, and then makes a recommendation on which one should be selected as the best UCL estimate. If the average or 95% UCL is greater than the maximum detected concentration, the maximum measured concentration was used as the exposure point concentration (EPA, 2002).

Appendix A provides the ProUCL output when there were sufficient samples to run statistics (soil and sediment). It should be noted that for avian receptors, the exposure point concentration was based on surface soil data because it is unlikely that the avian ROPC is exposed to subsurface soils given their habitat preferences, activities, and feeding behavior. One-half of the sample detection limit was used for sample measurements below the sample detection limit. There were not enough pond sediment or surface water samples for statistical calculations so average and maximum measured concentrations were used in the evaluation for these media.

Both averages and 95% UCLs were used in the SLERA to provide a range of exposure point concentrations. The dose estimates using the 95% UCL EPC were considered to represent reasonable maximum exposure (RME) and were the primary basis for conclusions, with the average EPC and resulting risk provided for supporting information. It should be noted, however, that neither average nor 95% UCLs were used in Section 2 to identify COPECS, and that exceedances shown on Figures 6 through 15 are based on point-by-point comparisons to screening levels. While hazard quotients for soil invertebrates and benthic organisms were estimated using average and 95% UCL concentrations, this was done to get an overall impression about site conditions and both evaluations were used to draw conclusions about site risks.

The general equation used for estimating COPEC dose from the soil/sediment and food ingestion pathways is presented below:

For a soil and sediment pathway:

$$Dose_{soil/sediment} = \frac{C_{soil/sediment} \times IR_{soil/sediment} \times AF_{soil/sediment} \times AUF}{BW}$$

For a food (dose) pathway:

$$Dose_{food} = \frac{C_{food} \times IR_{food} \times AUF}{BW}$$

Where:

$C_{soil/sediment}$	=	chemical concentration in soil/sediment (mg/kg)
C_{food}	=	chemical concentration in food (mg/kg)
$IR_{soil/sediment}$	=	soil/sediment ingestion rate (kg/day)
IR_{food}	=	food ingestion rate (kg/day)
$AF_{soil/sediment}$	=	chemical bioavailability factor from soil/sediment
(unitless)		
AUF	=	area-use factor (unitless)
BW	=	wildlife receptor body weight (kg)

It should be noted that the chemical bioavailability factor for all compounds in both soil and sediment was conservatively assumed to be 1 (i.e., 100% bioavailable for uptake). COPEC concentrations in food were estimated from soil/sediment concentrations using bioaccumulation factors (BAFs) or biota-sediment accumulation factors (BSAFs) with the following equation:

$$C_{food} = C_{soil/sediment} \times \text{BAF (or BSAF if sediment)}$$

For those receptors exposed through both soil or sediment and dietary exposure routes, the dose was assumed to be additive with the equation:

$$Dose_{total} = Dose_{soil/sediment} + Dose_{food}$$

Various literature sources, including the Wildlife Exposure Factors Handbook (EPA, 1993), were reviewed to determine the types and amounts of prey ingested by the wildlife receptors. Appendices C through I provide detailed intake (dose) calculations for each media and all receptors.

3.3 TOXICITY REFERENCE VALUES

Species-specific toxicity reference values (TRVs) were determined using scientific literature and other available resources with selected benchmarks generally based on measurements of survival, growth or reproduction in the laboratory. A TRV was selected from the available scientific literature for each compound using the following criteria (EPA, 1997):

- Doses based on the receptor species selected for evaluation were used preferentially; however, if toxicity information was not available for the species, doses for animals within the same class as the receptor species were used.
- Data for reproductive or developmental effects were used preferentially over other endpoints. Reproductive and developmental effects represent a more sensitive measure of wildlife effects than mortality. Therefore, these effects were chosen in preference to the less sensitive mortality endpoint for assessing ecological risk to the ROPCs.
- Chronic data were used preferentially to sub-chronic or acute data, and no observable adverse effects levels (NOAELs) were used in preference to lowest observable adverse effects levels (LOAELs) and effects measurements.

ERL values were used as sediment TRVs for benthic receptors. If the HQ was greater than 1 for a given compound, an alternate HQ was calculated using the ERM and the midpoint between the ERL and ERM to provide additional information about potential ecological risks to benthic receptors. In several instances, an Apparent Effects Threshold (AET) was used as the TRV because an ERL or ERM was not available. TRVs were not available for each receptor class or for each compound. Where appropriate, surrogate values were used within some chemical classes (e.g., DDT for DDE) for chemicals without TRVs but no species to species extrapolations were conducted per EPA (2009a). Because using surrogate values introduces considerable uncertainty into the risk assessment process, care was taken to only use surrogate values for chemicals with similar chemical structures or toxicities to minimize the uncertainty. The chemicals with no TRVs are discussed in the uncertainty section.

3.4 SCREENING-LEVEL HAZARD QUOTIENTS

The purpose of the risk characterization is to integrate the exposure and ecological effects analyses to determine if ecological receptors at the Site are potentially at risk from chemical exposure. In this section, the dose estimate is compared to the TRV to evaluate the potential for adverse health effects to the ROPC using a hazard quotient (HQ) approach. The HQ is a ratio of the estimated exposure concentration to the TRV where:

$$HQ = Dose / TRV$$

If the HQ is less than one, indicating the exposure concentration or dose is less than the TRV, adverse effects are considered highly unlikely. If the HQ is equal to or greater than one, a potential for adverse effects may exist. It should be noted that an HQ greater than one by itself does not indicate the magnitude or effect nor does it provide a measure of potential population-level effects (Menzie et al., 1992), and certainly should be evaluated based on the conservative nature of the assumptions. HQs were calculated for individual PAHs as well as for total PAHs, LPAHs, and HPAHs. PAHs were classified as LPAH or HPAH according to Box 3-6 of TCEQ guidance (TCEQ, 2001).

Tables 24 and 25 provide a summary of the HQs that exceed one for soil and sediment, respectively, for each receptor and COPEC. Section 3.4.8 discusses potential risks from surface water since HQs were not calculated for each sample that exceeded the surface water quality standard.

Appendices C through I provide the complete set of calculations for all compounds. A discussion of the results for each compound with a HQ greater than one follows for each media.

3.4.1 South Area Soil

As shown in Table 24, the RME NOAEL-based HQs for 4,4'-DDD and zinc exceed one for the earthworm receptor. The RME NOAEL-based HQs for antimony, zinc, and Aroclor-1254 for the Least shrew and zinc for the American robin exceed one.

3.4.2 North Area Soil

As shown in Table 24, the RME NOAEL-based HQs for zinc exceed one for the earthworm, Least shrew, and American robin receptors. The RME NOAEL-based HQs for antimony exceed one for the deer mouse and Least shrew receptors.

3.4.3 Background Area Soil

As shown in Table 24, the RME NOAEL-based HQs for barium and zinc exceed one for the earthworm and American robin receptors. The RME NOAEL-based HQs for antimony and zinc exceed one for the Least shrew.

3.4.4 Intracoastal Waterway Sediment

As shown in Table 25, the ERL-based HQs using the RME EPC for 4,4'-DDT, benzo(a)anthracene, dibenz(a,h)anthracene, fluorene, gamma-chlordane, phenanthrene, HPAHs and total PAHs exceed one for the benthic receptor. The midpoint between the ERL/ERM-based HQ for dibenz(a,h)anthracene using a 95% UCL was 1.3 while the average HQ was 0.3; none of the other compounds or HPAHs or total PAHs exceeded the midpoint of the ERL/ERM on a point-by-point comparison or when using the average or 95% UCL concentration as the EPC. As shown in Figure 9, dibenz(a,h)anthracene was measured in one sediment sample collected from the Intracoastal Waterway above the midpoint between the ERL and ERM. It should be noted that this measurement was below the ERM, however. The only benchmark available for hexachlorobenzene was the AET, and both the average and RME HQs exceed one for benthic organisms. None of the other NOAEL-based HQs was above one for sandpiper or green heron.

3.4.5 Intracoastal Waterway Background Sediment

As shown in Table 25, none of the NOAEL-based HQs for any compound for any receptor exceeds one even though, on a point-by-point basis, arsenic and nickel exceeded their ecological screening level at several sampling locations.

3.4.6 North Area Wetlands Sediment

As shown in Table 25, the ERL-based HQ using the RME EPC for many individual PAHs, 4,4'-DDT, endrin aldehyde, gamma-chlordane, zinc, LPAH, HPAH, and total PAHs exceed one for the benthic receptor. There is not an ERL for benzo(g,h,i)perylene or indeno(1,2,3-cd)pyrene. The AET-based HQs for benzo(g,h,i)perylene and indeno(1,2,3-cd)pyrene were 1.1 and 1.3, respectively, for the RME benthic scenario, although the HQs for the average scenario for both compounds were 0.3. The midpoint between the ERL/ERM-based HQ for dibenz(a,h)anthracene using a 95% UCL was 6.8 and the HQ based on the average concentration was 1.3. The midpoint between the ERL/ERM-based HQ for total PAHs using a 95% UCL was 1.2 and the HQ based on the average total PAH concentration was 0.2. None of the other compounds or LPAHs exceeded the midpoint of the ERL/ERM using the average or 95% UCL concentration as the EPC. The ERM-based HQs for dibenz(a,h)anthracene are 4.15 for the RME benthic scenario and 0.77 for the average benthic scenario. None of the NOAEL-based HQs exceed one for the sandpiper or green heron.

As shown in Figure 9, a point-by-point comparison indicates that several compounds are measured in individual samples above the midpoint of the ERL/ERM (highlighted in yellow). These exceedances include: 2-methylnaphthalene, acenaphthylene, chrysene, dibenz(a,h)anthracene, gamma-chlordane, lead, phenanthrene, pyrene, zinc, and HPAHs. Measured concentrations of chrysene, dibenz(a,h)anthracene, lead, zinc, and HPAHs exceed the ERM in at least one sample as shown in Figure 9 (highlighted in orange).

3.4.7 Pond Sediment

As shown in Table 25, the ERL-based HQs for 4,4'-DDT and zinc exceed one for the benthic receptor. The midpoint of the ERL/ERM and the ERM HQs for zinc exceed one for the RME benthic scenarios. The NOAEL-based HQs for zinc slightly exceed one for the RME sandpiper and green heron receptors (both are 1.3) but not for the average scenario for these receptors.

As shown in Figure 10, a point-by-point comparison indicates that zinc was measured in three samples above the ERM and the midpoint of the ERL/ERM. All three samples with zinc measured above the ERM were collected from the Small Pond.

3.4.8 Surface Water

A hazard quotient risk approach was not used to evaluate the surface water data since there are few toxicity values that allow HQs to be calculated that are protective of effects from dietary exposure. All compounds that were measured in excess of the screening criteria in Section 2.6 are listed below, and additional toxicity information is presented as well. Additionally, COIs that are considered bioaccumulative (4,4'-DDD, mercury, selenium and thallium) and that were measured above sample detection limits in surface water are also discussed below. Conclusions of risk based on exceeding screening levels are discussed in more detail in Section 5.0.

Acrolein. Acrolein was measured in one of four surface water samples collected in the wetlands area at a concentration of 0.00929 mg/L. It was not detected in any surface water samples from the Intracoastal Waterway or the two ponds. The single detection is greater than the TCEQ ecological benchmark value of 0.005 mg/L by less than a factor of two. There is neither a TSWQS nor a recommended national water quality criterion from the EPA (2009b) for chronic marine exposures.

Copper. The maximum measured concentration of dissolved copper in surface water collected from the wetlands area was 0.011 mg/L. It was not detected in any surface water samples from the Intracoastal Waterway or the two ponds. The maximum concentration is greater than the TSWQS of 0.0036 mg/L by about three-fold. The mean of the three detections is less than the TSWQS.

4,4'-DDD. The maximum measured concentration of 4,4'-DDD in surface water collected from the background area of the Intracoastal Waterway was 7.62 x 10^{-6} mg/L. It was not detected in any Site-related surface water samples. The two detections are less than the TCEQ ecological benchmark value of 2.50 x 10^{-5} mg/L, the maximum concentration being about three times less.

4,4'-DDT. The maximum measured concentration of 4,4'-DDT, and the only detection, in surface water collected from the background area of the Intracoastal Waterway was 1.30 x 10^{-5} mg/L. It was not detected in any Site-related surface water samples. The detection is about 13-fold greater than the TSWQS of 1.00 x 10^{-6} mg/L.

Mercury. The maximum measured concentration of total mercury in surface water collected from the wetlands area was 7.00 x 10^{-5} mg/L. It was not detected in any of the surface water samples from the Intracoastal Waterway or the two ponds. The two detections are less than the TSWQS of 0.0011 mg/L, the maximum concentration being about 16 times less.

Selenium. The maximum measured concentrations of total selenium in surface water collected in the Intracoastal Waterway and ponds were 0.063 and 0.0098 mg/L, respectively. It was not detected in the surface water samples from the background area of the Intracoastal Waterway or the wetlands. In the Intracoastal Waterway, the maximum and mean concentrations are about two-fold less and three-fold less, respectively, than the TSWQS of 0.136 mg/L. In the ponds, the single detection is about 14 times less than the TSWQS.

Silver. The maximum measured concentrations of dissolved silver in surface water collected from the Intracoastal Waterway background area and the ponds were 0.0058 and 0.0029 mg/L, respectively. It was not detected in the surface water samples from the Site-related area of the Intracoastal Waterway or the wetlands. In the Intracoastal Waterway background area, all detections are greater than the TCEQ ecological benchmark value of 0.00019 mg/L, the maximum being about 31 times greater. In the ponds, all detections are greater than the TCEQ ecological benchmark value, the maximum being about 15 times greater. There is neither a TSWQS nor a recommended national water quality criterion from the EPA (2009b) for chronic marine exposures. The TCEQ ecological benchmark value is derived from the EPA (2009b) acute marine recommended water quality criterion divided by a safety factor of 10.

Thallium. The maximum measured concentration of total thallium in surface water collected from the ponds was 0.0077 mg/L. It was not detected in any surface water samples from the Intracoastal Waterway or the wetlands. The two detections are less than the TCEQ ecological benchmark value of 0.0213 mg/L, the maximum concentration being about three times less. There is neither a TSWQS nor a recommended national water quality criterion from the EPA (2009b) for chronic marine exposures.

The maximum measured concentration of dissolved thallium in surface water collected from the ponds was 0.0032 mg/L. It was not detected in any surface water samples from the Intracoastal Waterway or the wetlands. All detections are less than the TCEQ ecological benchmark value of 0.0213 mg/L, the maximum concentration being about seven times less and the minimum concentration being about 15 times less. There is neither a TSWQS nor a recommended national water quality criterion from the EPA (2009b) for chronic marine exposures.

4.0 UNCERTAINTY ANALYSIS FOR STEPS 1 AND 2

This section describes the uncertainties associated with the methodology and results of the SLERA. Risk assessments (both ecological and human) necessarily require assumptions and extrapolations within each step of the analysis and this can lead to uncertainty in predicted risks. These uncertainties are generally the result of limitations in the available scientific data used in the exposure and risk models as well as their applicability to the Site. Accordingly, the key assumptions and uncertainties are thought to have the greatest influence on the ecological risks predicted for the Site and, as such, they are presented with a qualitative description of how the uncertainty may affect the evaluation and conclusions. This provides the risk manager with the appropriate context for understanding the level of confidence with the risk assessment results.

There are two principle sources of uncertainty – those resulting from natural variability and those resulting from data limitations. Both types of uncertainty are discussed as they relate to the three major steps of the SLERA: exposure assessment, effects characterization, and risk characterization.

4.1 EXPOSURE ANALYSIS UNCERTAINTY

This section primarily focuses on the uncertainties in the exposure analysis resulting from data limitations. There are three general categories of uncertainty that are discussed in this section: general exposure analysis uncertainties, receptor-specific uncertainties (i.e., uncertainties that are related to the receptors evaluated), and chemical specific uncertainties.

4.1.1 General Exposure Analysis Uncertainties

General exposure analysis uncertainties are those components of the exposure analysis that have not been or could not be well characterized for the assessment endpoints evaluated. Due to the conservative nature of the SLERA, it is believed that the overall impact of uncertainties related to the exposure analysis result in an overestimate of risk.

Data collected at the Site satisfied the goals described in the Work Plan (PBW, 2006a) and, thus, adequately characterized the Site's nature and extent of contamination. As described in the NEDR (PBW, 2009a), hundreds of samples of soil, sediment, and surface water were collected for the South Area, North Area, Intracoastal Waterway, and background soil, sediment, and surface water locations. Characterization was conducted for the entire Site and continued if a screening level was exceeded.

Overall, the data were determined to be of high quality. Data were collected and analyzed in accordance with approved procedures specified in the RI/FS Field Sampling Plan (PBW, 2006b) and were validated in accordance with approved validation procedures specified in the Quality Assurance Project Plan (QAPP) (PBW, 2006c). Very few of the data for any of the analytes were found to be unusable (ie., "R-flagged"). In instances where data were unusable, the analysis was conducted again (when possible) and the R-flagged datum was not used. Some of the data are qualified (ie., "J-flagged") as estimated because the measured concentration is above the sample detection limit but below the sample quantitation limit and/or due to minor quality control deficiencies. According to the *Guidance for Data Useability in Risk Assessment (Part A)* (EPA, 1992b), data that are qualified as estimated should be used for risk assessment purposes. Data quality was discussed in greater detail in the NEDR (PBW, 2009a).

In light of the thoroughness of the site characterization and because of the high quality data, it is believed that the calculated average and 95% UCL of the mean values accurately represent Site concentrations for chronic exposure conditions, such as those assumed in this evaluation, and that little uncertainty was incurred in the assessment due to incomplete site characterization. Organisms with home ranges smaller than the Site such as the earthworm and deer mouse for terrestrial receptors and *Capitella capitata* and fiddler crab for aquatic/estuarine receptors may be exposed to a locally higher concentration than the mean or 95% UCL. However, since the assessment endpoint is based on community survival and productivity and not individual survival and productivity, it is acceptable to use summary statistics to represent community risks. It should be noted, however, that a point-by-point comparison was also done to evaluate localized effects for the soil invertebrates and benthic receptors.

To assess impacts for groups of PAHs, such as total PAHs, LPAHs, and HPAHs, averages, maximums and 95% UCLs were identified for each individual PAH and added to derive a total PAH, LPAH, or HPAH average, maximum or 95% UCL for the class of compounds. This likely imparts unnecessary conservatism into the hazard quotient calculation because it assumes that the maximum measurement (or average, or 95% UCL) for every PAH falls within the same sample. Total HPAH, LPAH, and HPAH calculations were also conducted for each sample to ensure that an exceedance on a sample-by-sample basis was not inadvertently excluded from further evaluation.

The assumptions regarding ecological exposure on the South Area of the Site pose a highly conservative bias given that it was assumed that wildlife populations use and are exposed to the entire Site, and that these areas provide sufficient cover and/or foraging habitat to support these wildlife populations. The South Area was developed for industrial purposes and contains limited natural vegetative cover characteristic of viable ecological habitat. In many portions of the South Area, ground surface is covered by concrete slabs or the soil has been worked and there is a permeable cover such as gravel and/or oyster shell base that prevents nesting and foraging by many bird species, primarily insectivores and seed eaters. It should be noted, however, grasses and sparse weedy cover have grown since the operations at the Site have stopped, but this is a relatively small area when compared to the approximate 20-acre South Area. The developed and disturbed nature of the habitat at the South Area was not taken into consideration in the SLERA and, as such, risks are most likely overestimated for all receptors.

The same general uncertainty as described above applies to the risks associated with sediment from the Intracoastal Waterway since the area of the Intracoastal Waterway near the Site does not provide suitable habitat to encourage or keep fish and other ecological receptors at the Site as noted by USFWS (USFWS, 2005a). This conclusion was supported by observations during the fish sampling program when it took several weeks to catch the required number of fish (27) in the Intracoastal Waterway at the Site using gill nets. Fish were more plentiful (and thus more readily caught) in the background area that contained a higher quality habitat.

4.1.2 Receptor-Specific Uncertainties

Receptor-specific uncertainties include those parameters in the dose equation that have not been directly measured for receptors at the Site. Receptor-specific uncertainties applicable to both terrestrial and aquatic/estuarine receptors include the body weights and food and soil/sediment ingestion rates used to quantify exposure estimates. Often, the incidental soil/sediment ingestion rate was assumed to be a fraction of dietary intake since an alimentary study was not available to describe soil/sediment ingestion. All receptors were assumed to have an incidental soil/sediment ingestion rate of 2% although, per EPA (2009a), the American robin and Least shrew were assumed to have a 5.2% and 8% incidental soil ingestion rate. Additionally, dietary fractions of all receptors were based on literature data. Many of the receptors evaluated in the SLERA, such as the deer mouse and American robin, have been reasonably well studied so this was not considered a major uncertainty.

Per EPA guidance (EPA, 1997), it was assumed that the area use factor for all receptors was 100%, which most likely overestimates exposure and risk for the more mobile receptors such as the red-tailed hawk, coyote, sandpiper, and green heron particularly given the small size of the Site relative to the home range of these species. The conservatism is compounded with receptors that consume prey items since it was assumed that 100% of their prey comes from the Site as well.

Additional uncertainty may have occurred due to the species chosen to represent a guild and potential differences in their exposure patterns. It is believed, however, that the species chosen as the ROPC in the evaluation is similar enough to other species within a guild so that all are protected in the risk assessment process. It is difficult to predict the impact this uncertainty may have on overall risk predictions and conclusions.

4.1.3 Chemical-Specific Uncertainties

Chemical-specific uncertainties are those factors that are assumed for specific chemicals and generally relate to fate and transport modeling. These uncertainties should be considered in weighing the importance of the predicted risks for that chemical.

Bioaccumulation factors and biota-sediment accumulation factors were selected from available literature as noted in the toxicity tables provided in the appendices. They were not available for several of the compounds, and often the data available were sparse or of unknown quality. This makes assessing food chain effects in the evaluation difficult and sometimes uncertain. When appropriate, surrogate values for different chemicals and/or different receptors were used to allow for risks to be estimated for higher trophic level receptors when a chemical-specific value was not available. This approach imparts uncertainty into the evaluation although it is difficult to discern whether it leads to an over-estimation or under-estimation of potential risks.

Per EPA (2009a), if a bioaccumulation factor was not available and an appropriate surrogate could not be identified, a value of 1 was used to allow for the compound to be included in the food chain calculations. This likely leads to an overestimation of risk since many bioaccumulation factors are much less than one.

Bioavailability was assumed to be 100% per EPA guidance (EPA, 1997) although it is well known that metals and some organic compounds are less than 100% bioavailable (EPA, 2007b). This assumption leads to an overestimation of risks, which can be significant.

4.2 EFFECTS CHARACTERIZATION UNCERTAINTY

This section describes the assumptions inherent to the use of chemical-specific TRVs for chemicals evaluated in the terrestrial and aquatic/estuarine systems and chemical-specific ERLs/ERMs for chemicals evaluated for sediment-dwelling benthic organisms. PAHs in sediment, as discussed prior, were also evaluated as a class (total PAHs) and as subclasses (LPAHs and HPAHs).

Most available toxicity data were for standard laboratory animals or domestic animals such as rats, mice, quail, and mallards. Thus, these animals were used as surrogates to represent the toxicity of chemicals to site-specific receptors. It is unknown how the sensitivities of these surrogate organisms to toxicants compare to the sensitivities of the wildlife receptors evaluated at the Site. Using surrogate TRVs, therefore, may over- or underestimate toxicity and estimated risk to receptors at the Site.

Per EPA comments (EPA, 2009a), cross-class extrapolations are not advised and, therefore, the risk estimates for reptiles, crustaceans, and fish previously provided in the revised Draft SLERA (PBW, 2009b) were removed. The screening levels used to evaluate surface water and sediment are protective of the fish and crustaceans so this screening level risk assessment included these receptors, by design. Reptiles, however, were not evaluated in a quantitative manner. However, the following qualitative approach was followed to ensure protection to this receptor guild. There is not qualitative toxicological information that indicates source-related chemicals specifically produce greater toxicity to reptiles than to other guilds evaluated. Snakes have been observed at the Site and it is very likely that there are food resources available to support a snake population although the habitat at the South Area is not ideal. The terrestrial areas of the North Area likely provide ideal habitat for snakes although shallow groundwater may make subsurface conditions unfavorable for burrowing. It is unlikely that this receptor guild in more exposed or more at risk than the other receptors evaluated in the risk assessment.

The lack of screening values and toxicity data for several compounds imparts uncertainty on the evaluation although it is difficult to determine the significance of the uncertainty. It appears, however, that screening values and/or TRVs were available for the more toxic (relatively) and prevalent compounds (both frequency and concentration) at the Site.

The exception to this is for surface water. Many compounds measured in surface water did not have screening values, TSWQS, or EPA national recommended water quality criteria. Many of the compounds measured, however, are naturally occurring and all compounds were measured at relatively low concentrations. Uncertainties were evaluated for the eight COIs discussed in Section 3.4.8. Most of these COIs do not have chronic marine TSWQS or EPA recommended water quality criteria, namely acrolein, 4,4'-DDD, dissolved silver, dissolved thallium, and total thallium. Usually, lack of such standards or criteria is an indication that not enough is yet known about the toxic effects of the chemical or compound and/or the COI is classified by the EPA as a non-priority pollutant. Uncertainty, therefore, is associated with the benchmark value or screening level used in lieu of a better-researched standard or criterion. It follows, then, that conservatism would be purposely included in a benchmark value or screening level that would create an overestimation of potential risks. In particular, the ecological benchmark value for dissolved silver may be especially overly conservative because the value was derived by dividing the EPA national recommended water quality criterion for acute marine exposures by a safety factor of 10.

There are uncertainties in the PAH ERLs/ERMs used to assess risk to benthos. These values are based on effects to growth, survival, and/or benthic community indices for (largely) field collected sediments across the United States and should be used only as a screening tool (Long, et al., 1995). The use of field collected sediments imparts uncertainty in the establishment of these screening benchmarks and in any subsequent evaluation of sediment risk using these values because these sediments also contain concentrations of other chemicals that will affect sediment toxicity. The differences between the toxicity observed in the studies used to develop the ERLs/ERMs and site-specific measures of toxicity may be remarkable as observed at several site-specific studies where higher concentrations of PAHs did not result in toxicity (Alcoa, 2000 and Paine et al., 1996).

The AETs used to characterize risk for hexachlorobenzene, benzo(g,h,i)perylene, and indeno(1,2,3-cd)pyrene are based on screening sediment benchmarks developed for Puget Sound using a bivalve study, a Microtox assay, and a Microtox assay, respectively (Buchman, 2008). Sediment toxicity is highly variable based on local sediment conditions and, therefore, predictions of risk from screening values can vary greatly.

4.3 RISK CHARACTERIZATION UNCERTAINTY

This section discusses uncertainties related to the risk characterization and the methodology used to estimate risk. The most significant general uncertainty associated with risk characterization is how exposure to multiple chemicals was evaluated. Except for PAHs which are discussed below, additivity of effects to the various receptors from exposure to the multiple chemicals measured at the Site was not appropriate since these chemicals, for the most part, act via different mechanisms of toxicity. Furthermore, no evidence was found in the scientific literature to suggest that the toxicity of the compounds measured at the Site should be considered additive. Likewise, some metals are antagonistic but these effects were not considered either since the exact mechanism is not well understood toxicologically nor is there an accepted method for quantifying this type of interaction in the risk assessment.

For PAHs, potential effects were assumed to be additive and, as such, risks were estimated for total PAHs, LPAHs, HPAHs, and for individual compounds as well. This multi-pronged evaluation increases the confidence in the risk predictions as it provides for several lines of evidence to draw conclusions.

Background risks were estimated in a manner identical for site-related risks for soil and Intracoastal Waterway sediment, and qualitatively for Intracoastal Waterway surface water. Potential ecological risks from compounds measured in soil, as shown in Table 24, were very similar for site-related antimony and zinc when compared to the background area. Background area risks related via barium exposure to the earthworm and American robin were higher than site-related risks. Table 20 summarizes the statistical comparison to background concentrations for the various site media and indicate which compounds were measured in site-related media at concentrations statistically greater than background concentrations.

5.0 SUMMARY AND CONCLUSIONS OF THE SLERA

The SLERA can be used to assess the need and, if required, the level of effort required to conduct a baseline ecological risk assessment, or to determine that no further action is necessary. The SLERA can also be used to focus subsequent phases of the investigation by eliminating compounds from further evaluation (EPA, 2001). This section presents the summary and conclusions of the SLERA.

The screening-level ecological risk assessment evaluated the potential for unacceptable risk for terrestrial and aquatic/estuarine receptors as a result of direct (incidental ingestion) and indirect (bioaccumulation/biomagnifications through the food chain) exposure to chemicals measured in soil and sediment at the Site. Only direct toxicity to surface water was evaluated for the aquatic receptors as discussed herein.

Summaries of all soil and sediment HQs greater than one are provided in Tables 24 and 25 for soil and sediment, respectively, while Appendices C through I provide detailed risk characterization calculations for all compounds. Appendix J provides a list of all references cited in Appendices A though I. It should be noted that the risks presented below are conditional estimates of risk that should be taken in context with the discussion of uncertainty presented in the previous section.

5.1. Potential Ecological Risks Associated with Soil

Several of the risk calculations result in an RME HQ greater than one using the NOAEL as the TRV in soil from the South Area, North Area and background area, as shown on Table 24. The HQs for the other COPECS or receptors not listed in this table were below 1. Figures 6, 7, and 8 show a point-by-point comparison for compounds exceeding the screening criteria for the compounds listed in Table 24.

The RME HQs for antimony and zinc the earthworm, Least shrew, and American robin for all three areas (i.e., South Area, North Area, and background area) were similar and ranged from 1.4 to 14.9. The RME HQ for 4,4'-DDD in the South Area was 1.2 for the earthworm receptor while the RME HQ for Aroclor-1254 was 1.8 for the Least shrew. The RME HQs for the earthworm and American robin receptors in the background area were greater than 1 for barium as well.

5.2. Potential Ecological Risks Associated with Sediment

Figures 9, 10, 11, and 12 provide a sample-by-sample evaluation and show which compounds exceeding their screening criteria. This section also discusses the HQs estimated using average and 95% UCL concentrations for benthic receptors as well as the higher trophic-level receptors.

5.2.1 Intracoastal Waterway

As shown in Table 25, the ERL-based HQs exceed one for 4,4'-DDT, several individual PAHs, total HPAHs, total PAHs, hexachlorobenzene, and gamma-chlordane for the benthic receptor. None of the HQs for the sandpiper or green heron exceed one. Figure 9 shows a sample-by-sample comparison of compounds measured in excess of their benthic screening levels. No compounds were measured in excess of their ERM; dibenz(a,h)anthracene was measured at a concentration greater than the midpoint of the ERL/ERM in one of sixteen samples; and hexachlorobenzene was measured in the same sample at a concentration greater than the AET, which was the only available benchmark for that compound.

Hexachlorobenzene was detected in one of sixteen sediment samples collected in the Intracoastal Waterway. The concentration at sample location IWSE07 (0.0319 mg/kg) exceeded the AET (0.006 mg/kg), which suggests a possible risk for the benthic receptor. (Note that hexachlorobenzene is not listed in Figure 9 as there was not a TCEQ or EPA screening criteria.) This single hexachlorobenzene detection was "J" flagged, which means it was an estimated value below the sample quantitation limit. Hexachlorobenzene was not measured above the sample detection limit (which was specified in Appendix D of the approved QAPP (PBW, 2006c)) at an adjacent sample location approximately 40 feet from IWSE07 or at the three other sediment sample locations in the same barge slip.

Dibenz(a,h)anthracene was measured in one sample at location IWSE07 above the mid-point between the ERL/ERM as shown in Figure 9 by the yellow highlighting. Dibenz(a,h)anthracene was not measured above the sample detection limit (which was specified in Appendix D of the approved QAPP (PBW, 2006c)) at an adjacent sample location approximately 40 feet from IWSE07 or at the three other sediment sample locations in the same barge slip. None of the other PAHs, total PAHs, total HPAHs, or total LPAHs in sample IWSE07 exceeds the midpoint of the

ERL/ERM, although acenaphthene, fluorene and total HPAH exceed the ERL in this sample. The RME midpoint ERL/ERM-based HQ for the benthic receptor potentially exposed to dibenz(a,h)anthracene in sediments of the Intracoastal Waterway was 1.3.

None of the HQs for the sandpiper or green heron exceed one for the Intracoastal Waterway sediments.

5.2.2 Background Intracoastal Waterway

The only compounds that exceeded their screening level in sediment from the background Intracoastal Waterway area were arsenic and nickel, as shown in Figure 10. None of the HQs for these two compounds or the other COPECs identified in Table 21 exceeded one.

5.2.3 North Area Wetlands

As shown in Table 25, the ERL-based HQs exceed one for 4,4'-DDT, a number individual PAHs, total LPAHs, total HPAHs, total PAHs, endrin aldehyde, gamma-chlordane, and zinc for the benthic receptor. None of the HQs for the sandpiper or green heron exceed one. Figure 11 shows a sample-by-sample comparison of compounds measured in excess of their benthic screening levels. Orange highlighted concentrations on this figure are compounds measured in excess of their ERM, specifically dibenz(a,h)anthracene (in 5 of 48 samples), chrysene (in 1 of 48 samples), lead (in one of 48 samples), zinc (in 3 of 48 samples), and total HPAHs (in 3 of 48 samples). Yellow highlighted concentrations on this figure are compounds measured in excess of the midpoint of the ERL/ERM and include 2-methylnaphthalene (in 1 of 48 samples), acenaphthylene (in 2 of 48 samples), benzo(a)anthracene and benzo(a)pyrene (in 1 of 48 samples), chrysene (in 1 of 48 samples), gamma-chlordane (in 1 of 48 samples), phenanthrene (in 2 of 48 samples), pyrene (in 1 of 48 samples), total HPAHs (in 1 of 48 samples), and zinc (in 3 of 48 samples).

The ERL-based HQs for dibenz(a,h)anthracene ranged from 3.2 to 17.4 for the average and RME benthic receptor scenarios, respectively, which suggest that adverse benthic risks from North Area wetlands sediments are possible. So, while localized adverse effects may be possible at the sampling locations that exceed the mid-point of the ERL/ERM and the ERM as shown in Figure 11, it is difficult to estimate the potential significance of the impacts for the benthic community of the North Area wetlands, which is roughly 15 acres in size and is part of a wetlands system that covers hundreds of acres. Dibenz(a,h)anthracene is not considered bioaccumulative (TCEQ, 2001) and none of the risk estimates for the higher trophic-level receptors have HQs greater than one for this compound.

Evaluating the dibenz(a,h)anthracene data closer reveals that 6 of 48 samples exceeded the ERL while 5 of 6 samples exceeded both the ERM for marine sediment and the midpoint of the ERL and ERM. Three of these samples have total HPAH concentrations exceeding the ERL but the total LPAH and total PAH do not exceed the midpoint of the ERL/ERM or the ERM.

As noted in Section 2.0, there are no indications that the benthic community in these six locations is stressed or has been impacted by the dibenz(a,h)anthracene or other compounds present in the sediment. It is unclear why the toxicity value for this compound is significantly lower than the benchmarks derived for the structurally similar PAHs but it is clear that this low value significantly impacts risk perception and should be taken in context with other benchmarks.

While lead and zinc were measured in at least one sample at a concentration greater than the ERM, their midpoint of the ERL/ERM HQs were less than one, and none of the NOAEL-based HQs for either compound were above one for the sandpiper or green heron.

5.2.4 Ponds

As shown in Table 25, the RME ERL-based HQs for 4,4'-DDT and zinc were greater than one, specifically 1.6 for 4,4'-DDT and 6.7 for zinc. The RME ERM-based HQ for zinc is 2.4 while the RME NOAEL-based HQs for both the sandpiper and green heron receptors exposed to zinc in pond sediments are 1.3 and the average HQs are 0.4. Figure 12 shows each sample location where a compound was measured in excess of a screening level, and the associated concentration. At all three sampling locations in the Small Pond, zinc was measured at a concentration greater than the ERM. Zinc concentrations measured in pond sediments at the Small Pond are similar to zinc measured in soil at the background area, as shown on Figure 8.

5.3 Potential Ecological Risks Associated with Surface Water

Figures 13, 14, and 15 respectively show surface water concentrations of COPECs in the background Intracoastal Waterway, wetlands area, and ponds that were measured in excess of their screening levels. There is not a figure for Site surface water sampled collected from the Intracoastal Waterway since none of the compounds measured above detection limits in these samples exceeded its screening criteria.

5.3.1 Intracoastal Waterway

No compounds were measured in excess of their screening criteria in the Intracoastal Waterway surface water. Selenium (dissolved), which is considered bioaccumulative, was measured at a maximum and mean concentration roughly two and three times less, respectively, than the TSWQS. Even though selenium at the measured concentration range may not cause potential adverse effects to aquatic life, it is difficult to assess the likelihood of adverse risk to higher trophic-level receptors that prey on water-borne organisms.

5.3.2 Background Intracoastal Waterway

Dissolved silver and 4,4'-DDT were measured in surface water from the background area of the Intracoastal Waterway above their ecological benchmark value or TSWQS in at least one sample. In addition, 4,4'-DDD, which is considered bioaccumulative, was detected in surface water from the background area of the Intracoastal Waterway but at a concentration below the benchmark. Both 4,4'-DDT and 4,4'-DDD are considered bioaccumulative. It is difficult to assess, however, the likelihood of adverse risk to higher trophic-level receptors that prey on water-borne organisms that may be exposed to these compounds.

5.3.3 North Area Wetlands

Maximum concentrations of acrolein and dissolved copper were detected in at least one surface water sample from the wetlands area at concentrations that exceeded their respective ecological benchmark value or TSWQS. It is believed that there is insignificant risk from the presence of acrolein because of its infrequent detection (i.e., one of four samples) and the fact that its concentration is less than twice the ecological benchmark value. Although the maximum detected concentration of dissolved copper in a wetlands area surface water sample is greater than the TSWQS by about three-fold, the mean concentration of all four samples is less than the standard. Therefore, it was assumed that there is insignificant risk from the presence of copper in the wetlands area surface water. The maximum measured concentration of total mercury in the wetlands area is about 16 times less than the TSWQS. Even though mercury has bioaccumulative properties in aquatic and terrestrial food chains, it is believed that there is insignificant risk in the wetlands area because the maximum detected concentration is so much lower than the TSWQS.

5.3.4 Ponds

The maximum concentration of dissolved silver detected in a surface water sample from the ponds exceeded its ecological benchmark value. The maximum and mean concentrations of dissolved silver in the ponds are about 15 times greater and ten times greater, respectively, than the ecological benchmark value. However, the maximum and mean concentrations of dissolved silver in the surface water from the background area of the Intracoastal Waterway are more than twice the maximum and mean concentrations in the ponds. Additionally, the maximum and mean concentrations of dissolved silver in the surface water from the background area of the Intracoastal Waterway are about 31 times greater and 28 times greater, respectively, than the ecological benchmark value. Therefore, it is assumed that there is an insignificant risk from the presence of dissolved silver in the ponds.

Both selenium (total) and thallium (dissolved and total) were detected in surface water of the ponds at a concentration below the ecological benchmark. Both are considered bioaccumulative. It is believed that there is an insignificant risk in the ponds, however, because both were measured at concentrations so much lower than their ecological benchmark values.

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